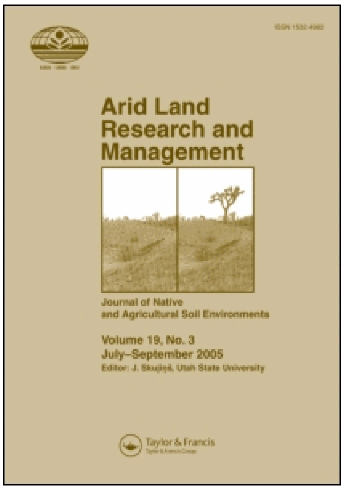

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Long-Term Ecological Monitoring

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The intent of long-term ecological monitoring is to document changes in important properties of biological communities. At the least, a long-term monitoring system should be designed to detect long-term trends in three key attributes: soil and site stability, hydrologic function, and the biotic integrity of the system. There are four basic guidelines for developing integrated soil-vegetation monitoring systems for rangelands. These are: (1) identifying a suite of indicators which are consistently correlated with the functional status of one or more critical ecosystem processes and/or properties; (2) selecting base indicators on site specific objectives and resource concerns, and inherent soil and site characteristics; (3) using spatial variability in developing and interpreting indicators to make them more representative of ecological processes; and (4) interpreting indicators in the context of an understanding of dynamic, nonlinear ecological processes. To the extent possible, indicators should reflect early changes in ecological processes and indicate that a more significant change is likely to occur. In addition to these guidelines, measurements included in long-term monitoring systems should be rapidly applied, simple to understand, inexpensive to use, and quantitatively repeatable.

Keywords indicators, rangeland health, biotic integrity, resilience, resistance

Monitoring in some fashion is required to evaluate progress towards meeting management objectives of any enterprise. For rangeland enterprises, some planned approach to monitoring is essential to any attempt to practice adaptive resource management. One of the major challenges facing rangeland resource managers is how to monitor vegetation structure and dynamics in a scientifically-rigorous, but cost-effective and widely-understood fashion (NRC, 1994). Many site-specific, field-based protocols are available to monitor structural characteristics of rangelands (e.g., NRCS, 1997; Elzinga et al., 1998). These monitoring approaches are effective for determining detailed vegetation status at specific locations within a site, and are often used to guide short-term management decisions. Methodologies which establish an "annual-use record" that provide information for management decisions on a seasonal or annual time frame can be labeled as short-term monitoring. Methodologies which generate information that create a "trend record" can be labeled as long-term monitoring. Documentation of dynamics such as soil surface structure and plant basal cover changes that are recorded every three to five years would be characteristic of long-term monitoring. Short-term and long-term monitoring programs can be effectively combined (Figure 1).

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Yet, use of currently available tools to assess and monitor rangelands can be limited by a combination of expense, training requirements, and poor repeatability. Unfortunately, there is often a poor correlation between measured indicators and the properties and processes of interest; in particular, an explicit consideration of rangeland function is often ignored (Herrick, 2000; Herrick et al., 2002). Existing approaches often rely almost exclusively on vegetation indicators, and they can be relatively insensitive to ecological thresholds that can limit the ability of rangelands to recover from perturbations (Friedel, 1991; Archer, 1994; Davenport et al., 1998). Rapid vegetation changes are often preceded by processes such as erosion or seed pool depletion that operate over relatively long periods (Sheffer et al., 2001). Therefore, monitoring indicators upon which they are based tend to reflect consequences of changes in ecosystem function, but not the underlying causes nor early warning signs of pending ecological transitions (Brown & Havstad, 2003).

Despite these real limitations, the intent of monitoring, especially long-term monitoring, is to document changes in important properties of biological communities. There are measurable indicators that should relate to ecological processes defining vegetation states or their trajectories of change, and these relationships should reflect system constraints that could be addressed by management practices. Any discussion of monitoring programs and their validity and application needs to address the nuances, characteristics, and limitations of indicators.

Identify Indicators that are Consistently Correlated with Function

An indicator is an observable component of a system which reflects an important property or process of that system. Indicators may be quantitative, qualitative, or

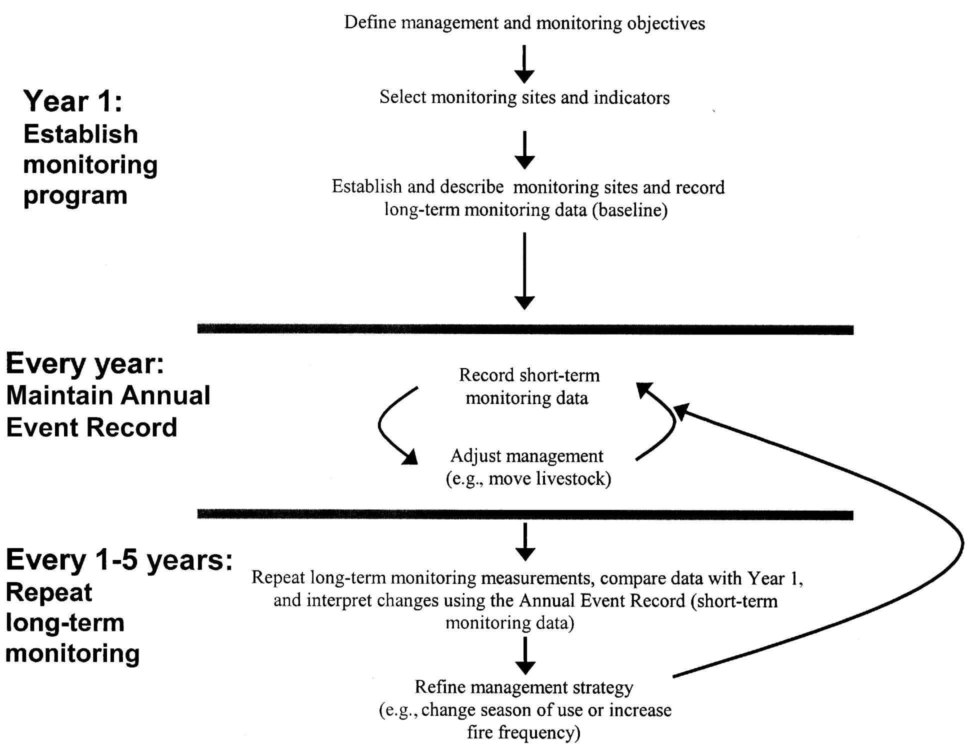

FIGURE 1 Integration of short-term and long-term monitoring programs with management (from Herrick et al., 2003).

both. Rangeland monitoring (and assessment) is dependent upon indicators, and the type of monitoring conducted, as a function of management objectives, is reflective of the indicators chosen. Indicators should add value to a monitoring program by providing information about the functioning of the system which cannot be derived directly from knowledge of the management system (Brown et al., 1998). Indicators need to be consistently correlated with some ecosystem function. An indicator is of little management value if it lags behind the process of interest. However, the indicators should also be reflective of actual changes in the system, rather than changes which are assumed to follow from changes in management.

Indicators for use on rangelands can be and have been based on previously published studies, new research, and expert knowledge about the variability in relationships across diverse rangeland ecosystems. Table 1 presents a few indicators derived from core and supplemental quantitative measurements. These indicators are calculated from three core measurements and a number of supplementary measurements (Table 1). Each of the core measurements can be used to generate a suite of indicators which are related to each of three attributes or criteria: soil and site stability, watershed function, and biotic integrity. The supplementary measurements can be applied depending on resource concerns and site characteristics.

The core measurements include line-point intercept, continuous line intercept, and an aggregate stability test. The line-point intercept is used to quantify plant cover and composition, and soil surface characteristics (Bonham, 1989). This measurement is used to generate a number of indicators including bare ground, which is highly correlated with both runoff and susceptibility to water erosion (Smith & Wischmeier, 1962; Blackburn & Pierson, 1994), and basal cover, which is related to overland flow path length and to the capacity of the system to recover following a severe disturbance. (Herbel et al., 1972; Gutierrez & Hernandez, 1996). Increasing overland path flow length increases the amount of time

TABLE 1 Selected Indicators and their Quantitative Measurements, and their Relevance to Each of Three Landscape Attributes (from Herrick et al., 2003)

Indicator(s)/measurement	Soil & site stability	Hydrologic function	Biotic integrity
Core			
1. Plant cover and composition using line point intercept	X	X	X
2. Canopy gaps using continuous line intercept (minimum 20 cm between canopy elements)	X	X	X
3. Soil stability test	X	X	X
4. Belt transects for woody & invasive plants	X	X	X
Supplementary			
5. Impact penetrometer		X	
6. Single-ring infiltration		X	X
7. Plant production by species (double sampling)		X	
8. Species richness			X
9. Vertical structure (cover pole)			X
10. Tree density	X	X	X
11. Riparian channel vegetation survey	X	X	X
12. Riparian channel and gully profile	X	X	X

available for infiltration to occur. Microbiotic crust cover can be calculated separately from the line-point intercept data for systems in which these organisms play an important role in stabilizing the soil surface (Eldridge & Kinnell, 1997; Belnap & Gillette, 1998).

Noncanopy patches larger than a minimum diameter (e.g., 20 cm) are recorded along a continuous line intercept. These patches cannot be detected using the line point intercept method and are highly correlated with susceptibility to wind and water erosion, and to the invasion of some species which change vegetation structure (Gould, 1982; Musick & Gillette, 1990). The size of canopy gaps is also an indicator of the relative uniformity of soil resource distribution (Schlesinger et al., 1990).

The third core method is a field aggregate stability test (Herrick et al., 2001). This test is used to rate water stable aggregation on a scale from one (slakes immediately) to six (75% remains on 1.5 mm screen following sieving) for soil surface fragments 6–8 mm in diameter. The method is highly correlated with laboratory measurements of aggregate stability (Herrick et al., 2001) which, in turn, have been negatively correlated with inter rill soil erosion in the field (Blackburn & Pierson, 1994). Aggregate stability, as determined by this method, is relatively insensitive to intensive short-term disturbances such as trampling by horses, humans, and vehicles, but reflects longer-term changes in soil structure. Insensitivity to single disturbance events is critical to ensure that monitoring results do not simply reflect normal variability in the system. Soil aggregate stability at the surface is particularly important in plant canopy interspaces when rock and litter cover are minimal because there is no protection from raindrop impact. We recommend calculating aggregate stability, as well as rock, microbiotic crust, and litter cover, separately for plant canopy and intercanopy spaces.

The belt transect, the final core method, is used for early detection of invasive species, both native and exotic. Many of these species, including mesquite (*Prosopis glandulosa* Torr.), juniper (*Juniperus* spp.), Lehmann's lovegrass (*Eragrostis lehmanniana* Nees), and cheatgrass (*Bromus tectorum* L.) have been shown to be associated with dramatic changes in soil quality, as reflected in changes in carbon and nutrient cycling processes (Barth & Klemmedson, 1982; Schlesinger et al., 1996; Connin et al., 1997; Arredondo & Johnson, 1999), soil erosion (Davenport et al., 1998), and infiltration capacity (Reid et al., 1999). However, the establishment of these species is often simply an indication that a change has already occurred (Brown & Archer, 1999), significantly reducing their value as indicators.

The eight supplementary methods (Table 1, 5–12) have also been correlated with ecosystem function. Species richness is a direct measure of the number of species present on a site, calculated using a species area curve based on counts in plots of different sizes (Stohlgren et al., 1995). Plant production using double sampling is normally used for assessment only as it is relatively imprecise and varies dramatically among years.

The Problem of Scale

Many ecological processes operate at a submillimeter scale, are observed at a centimeter scale, but are often managed at a scale of a kilometer or more. These processes may vary by several orders of magnitude at any or all of these scales. The degree to which spatial heterogeneity is expressed also depends on the scale at which observations or measurements are made (Pickett & Cadenasso, 1995). Consequently, the scale at which processes are most easily observed or conveniently managed is rarely the scale which is most relevant.

The problem of scaling up from processes which occur at a single point to pasture or landscape impacts is compounded by the patchiness of soil and vegetation (Schlesinger & Pilmanis, 1998) and by previous land uses (Blackmore et al., 1990).

Spatial variability in rangelands at individual plant, patch, plant community, and landscape scales has been well described, particularly for nutrients (Mazzarino et al., 1996; de Soyza et al., 1998), organic matter (Burke et al., 1995) and to a lesser extent for microbial activity (Herman et al., 1995; Mazzarino et al., 1996) and microarthropods (Cepeda-Pizarro & Whitford, 1989). Soil processes often vary by several orders of magnitude over time at intervals as short as several minutes in response to changes in resource availability (Lange et al., 1992). These changes in resource availability occur at multiple temporal and spatial scales (WallisDeVries et al., 1998), and are most obviously associated with external factors such as grazing (Kellner & Bosch, 1992).

The relationships between patterns of properties and processes are further complicated by biological feedbacks occurring within and among patches at different spatial and temporal scales. One potential solution to at least some of these problems is to use indicators which tend to integrate the effects of a variety of processes. Soil organic matter is a key soil property which reflects the status of a wide range of soil physical, chemical, and biological processes. The spatial distribution of soil organic matter therefore reflects the variability in these processes across different spatial scales. A comparison of the scales at which soil organic matter varies with the scales at which it is managed illustrates the problem of linking management with processes. For example, in a patchy arid grassland, incorporation of standing dead grass stems and leaves into the soil by the hoof action of grazing animals can enhance organic matter cycling. However, the identical disturbance by animal hooves in an adjacent interspace can dramatically reduce the capacity of the soil to absorb water and increase soil loss due to both wind and water erosion. This is just one situation in which a management tool applied at one scale (the pasture) can have dramatically different effects in different patches at the scale of an individual plant.

The spatial distribution and temporal pattern of processes can also be very important. Just as the simultaneous removal of multiple stresses in a particular microsite can facilitate seedling establishment (Grime, 1977), so also can the simultaneous application of stresses lead to degradation in areas in which sequential application of the same stresses would have little or no effect (Whitford et al., 1999a). For example, most arid and semiarid plants are adapted in some way to drought and tolerate grazing. However, when overgrazing is combined with drought, the resistance of the system to degradation is overwhelmed. These barriers can be overcome by (1) adopting integrative soil indicators which reflect the function of a range of ecosystem processes, (2) exploiting spatial variability in both monitoring and management, and (3) applying and interpreting these indicators in the context of variability at scales range from the microaggregate to macroattachments.

Base Indicator Selection on Resource Concerns and Site Characteristics

The core measurements listed in Table 1 can be used to generate management-relevant indicators in many situations at both local and regional scales. However, monitoring efficiency can often be increased by selecting only those supplementary indicators which are most sensitive to site-specific changes in ecosystem function, and which are relevant given the soil and site characteristics.

Given that societal values, as well as scientific understanding, change over time, monitoring programs should be designed to quantify the potential of the system (1) to function in support of a range of societal values rather than to support any individual value, (2) to resist degradation, and (3) to recover following degradation. The premise that the capacity of an individual site to continue to function depends on a core set of processes is common to most definitions of both soil quality and rangeland health. The core measurements were selected to generate indicators of these processes. Indicators which address specific values or land uses, such as livestock forage or wildlife habitat, can often be calculated from the core measurements (Table 1) and additional measurements can be included. In rangelands, however, it may be useful to maintain a distinction between core and supplementary measurements to minimize unnecessary cost increases. The primary criterion for indicator selection must be the strength and consistency of its relationship to a critical process, while recognizing that the relative importance of different ecological processes, and the strength of the relationship between indicator and process, varies among soils, landscape positions, and regions. In some cases, relationships between indicators and processes can be inferred from the literature, while in others, they must be quantified with new studies.

Integrative Indicators which Reflect Multiple Processes

It is unlikely that a single indicator exists which can reliably reflect changes in ecosystem function across rangeland systems or even within a single ecosystem (NRC, 1994). For example, specific organisms such as ants may be extremely useful in some situations, such as mineland rehabilitation (Majer, 1983). However, specific organisms, such as ants, may be far too variable for broad use as indicators for assessment or monitoring of rangelands (Whitford et al., 1999b). It should be possible to identify a suitable suite of rapid, repeatable, cost-effective measurements which together reflect a variety of processes and functions. Key soil processes are nearly impossible to measure directly. Indicators of soil processes fall into two categories: "input" indicators which reflect the conditions necessary for the process to occur (e.g., presence of organic matter inputs for decomposition, mineralization, formation of soil aggregates, etc.) and "output" indicators which suggest that a process has occurred in the past (e.g., nutrient availability). Some indicators, such as pore size distribution, fall into both categories. Simple input-type predictive indicators are rare because biological processes are generally controlled by more than one factor, and it is impossible to measure all factors. The application of indicators which reflect historic function is limited by the fact that it is usually impossible to know how long ago a process was functioning, or whether it is still functioning. Indicators which reflect more recent activity tend to have a higher signal to noise ratio compared to those which reflect cumulative historic activity. For example, microbial biomass carbon is a reasonable indicator of a system's capacity to convert organic inputs into soil organic matter. However, this ability varies with season, recent weather patterns, and interactions with soil flora and fauna. It also varies with substrate availability. High microbial biomass carbon often simply reflects a recent organic matter input, such as leaf fall from deciduous shrubs or roots from annuals. Conversely, trends in the humic fraction tend to be insensitive to short-term weather patterns, but are also unlikely to reflect short-term management-induced changes. In some cases, it is possible, if not likely, that there are management associated differences, but sampling and measurement-induced variability masks them. Some sensitive indicators which have a low signal to noise ratio because of weather and season can still be used if a local reference site is available for comparison. Ideal indicators for early detection of ecosystem change are those which reflect the status of a variety of processes, are temporally stable and, most importantly, reflect susceptibility of the system to degradation (resistance) or capacity to recover (resilience) if degradation occurs.

A Multi-scale Approach to Monitoring

Most monitoring programs are based on measurements made at individual points. This approach misses many of the important processes which are only expressed at coarser scales. Furthermore, a point-based approach to monitoring diverse rangeland ecosystems frequently leads to impractical and unaffordable numbers of replications. The use of emergent properties of landscapes is often suggested as a way to integrate multiscale processes and identify changes in these processes before a threshold is reached. Some properties, such as sediment loads and stream hydrographs, are relatively straightforward to interpret when appropriate reference data are available. More frequently, however, these properties (such as spatial organization of aggrading and degrading areas) are difficult to identify and even more challenging to quantify. Even when they can be quantified in one landscape, it is often difficult to develop tools, such as filters and algorithms for pattern analysis, which can be applied systematically across a variety of landscapes.

A flexible approach to rangeland monitoring is required in response to the challenge of providing management information applicable at the landscape level while measuring patterns of properties (reflecting processes) at the point scale and incorporating qualitative information extracted from multiple scales. Qualitative soil and vegetation indicators are used together with information about soils, geomorphology, and current and historic vegetation and land use patterns to stratify the landscape. This information is then interpreted in the context of monitoring objectives to select specific monitoring points and indicators, and to establish appropriate monitoring frequencies for quantitative measurements.

Interpreting Indicators Based on an Understanding of Ecological Processes

Significant progress has been made in the identification of suitable indicators for rangeland ecosystems (Doran & Parkin, 1994, 1996; NRC, 1994; Brown et al., 1998), but integration and interpretation of these indicators has been more difficult. A number of approaches have been suggested and successfully applied to some systems. Indices have been developed from linear combinations of indicators (e.g., Doran and Parkin, 1994), and these approaches are useful for documenting change in systems which are gradually evolving. However, ecological theory suggests that a more dynamic model may be more appropriate in systems which are structured by relatively infrequent catastrophic disturbances, or in which cumulative effects are not expressed until a threshold is reached (Holling, 1973).

This concept has been applied in rangeland ecosystems in the form of state and transition models (Westoby et al., 1989; Friedel, 1991). A unique state and transition model can be described for each soil, or suite of similar soils. For example, the state and transition diagram illustrated in Figure 2 is based on current understanding of the ecological dynamics on a northern Chihuahuan Desert site with sandy soils (Bestelmeyer et al., 2003). This diagram is relatively simple: community pathways branch within states, but there are at most two potential transitions for each state. The number of potential states could easily increase in response to climate change, or to species invasions. Earlier state and transition models often recognized individual communities as separate states, whether or not they were separated by thresholds, making it difficult to consistently define states.

The key points illustrated by this example are that transitions are defined by nonlinear changes in the function of the system, and that mean values of a suite of indicators may provide relatively little information about the status of the system during irreversible transitions. For example, herbaceous species composition and productivity are commonly considered to be good indicators of condition. However, Northrup & Brown (1999) illustrated that while a tussock-grass dominated community appeared to be stable based on these indicators, loss of biological integrity in response to overgrazing was occurring via the fine-scale redistribution of soil resources. In another example, Brown & Ash (1996) showed

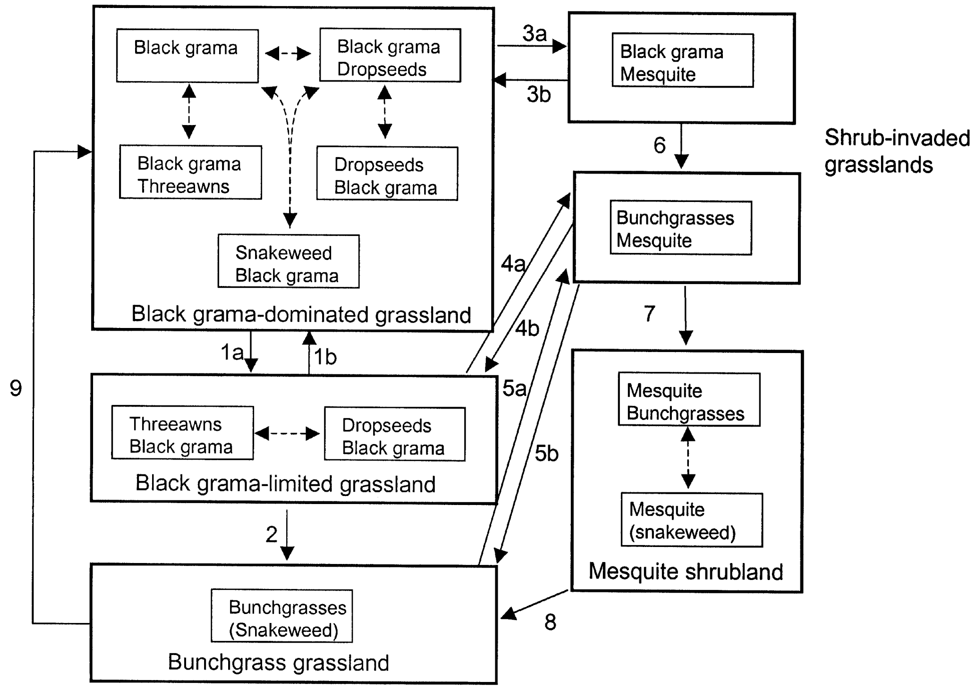

FIGURE 2 Example of a state-and-transition model for a sandy ecological site within the SD-2 land resource unit of southern New Mexico. Dashed arrows within boxes (states) represent easily reversible plant community composition and associated soil changes. Unidirectional solid arrows between states reflect that the transitions are nonreversible without external inputs. The diagram has been simplified and does not include all possible plant communities in all states (from Bestelmeyer et al., 2003).

that monitoring species composition and plant production data to detect shrub invasion can result in delaying important management decisions until well after woody invaders have become established. Indicators which reflect transitions between states of ecological sites need to be identified, evaluated, and calibrated for use in long-term monitoring.

Conclusions

Although the importance of integrating soil and vegetation indicators is widely acknowledged as integral to capturing ecological dynamics (NRC, 1994; West et al., 1994), few indicators for long-term monitoring are currently available that can be cost-effectively applied on rangelands, and even fewer have been rigorously tested. Most monitoring approaches were designed to compare the vegetation cover and composition of an area with its historical reference or to determine the ability of an area to support livestock production; thus these approaches do not reflect societal demands for a broad variety of land uses (NRC, 1994). Furthermore, many approaches are based on data collected at individual points or locations. Conducting extensive field surveys to cover the range of vegetation and soil types found within a site is labor- and time-intensive. Point-based sampling also ignores interactions among different parts of a landscape, such as the redistribution of water from upslope to down slope locations. Although remote sensing tools can cover larger areas, the resolution is often too coarse to examine connections among locations.

There are four basic guidelines for developing integrated soil vegetation monitoring systems for rangelands. These are (1) identifying a suite of indicators which are consistently correlated with the functional status of one or more critical ecosystem processes and/or properties, (2) selecting base indicators on site specific objectives and resource concerns, and inherent soil and site characteristics, (3) using spatial variability in developing and interpreting indicators to make them more representative of ecological processes, and (4) interpreting indicators in the context of an understanding of dynamic, nonlinear ecological processes.

There are several important research areas that need to be addressed in order to more effectively develop long term monitoring techniques based on valid indicators:

(1) Cost-effective vegetation and soil measures are needed that more accurately reflect the status of underlying ecological processes (NRC, 1994). Predictive indicators should reflect the long-term sustainability of the system. Such indicators must be directly calibrated to ecosystem functions or indirectly calibrated using simulation models to evaluate the relative sensitivity of various indicators to changes in ecosystem functions. Indicators also should be sufficiently general to apply to many different types of dry and mesic rangelands, including grasslands, shrublands, woodlands, and savannas. Individual vegetation and soil indicators need to be well-integrated both spatially and temporally, and their relationships to landscape fluxes need to be recognized. Measures should be easy to use, understand, and interpret (Sundt, 2002).
(2) Remote sensing tools must be integrated with ground-based measurements to detect change at management-relevant scales. While the remote sensing community has focused on developing stand-alone applications and most field ecologists have restricted themselves to point measurements, hydrologists have successfully integrated the two approaches (Rango et al., 1983). Remote sensing is the most efficient and effective way to stratify the landscape into relatively homogeneous monitoring units, but its ability to detect ecologically significant changes is limited by several factors. Resolution of remotely-sensed data is often too coarse to detect fine-scale changes in pattern. These types of data are often insensitive to changes in soil properties and processes. New approaches, such as diversity indices (e.g., Tanser & Palmer, 1999), will also be needed to better reflect changes in vegetation structure as multiple spatial scales are integrated.
(3) Conceptual and simulation models need to be developed and integrated with the remote sensing and ground-based measures in order to evaluate the sensitivity of different parts of the landscape to both new and existing threats and management changes. This information can focus monitoring efforts on sensitive areas in order to use quantitative data effectively for adaptive management (Puigdefabregas, 1995). Often, monitoring protocols do not conceptually integrate relationships among ecological, climatic, and anthropogenic drivers (Brown & Havstad, 2003). State and transition models could provide a needed conceptual basis by identifying the range of possible changes in vegetation associated with multiple drivers and soil constraints (Bestelmeyer et al., 2003). Multiple sources of empirical data can be related together in these models. Furthermore, state-and-transition models distinguish those changes that are easily-reversible from those that are relatively persistent, thus providing a basis for quantifying resistance and resilience (Stringham et al., 2003; Bestelmeyer et al., 2003). Simulation models supported by empirical studies of threshold behavior can be used to provide this quantitative approach to predicting site sensitivities based on drivers such as vegetation-soil feedbacks, herbivory by domestic and native animals, and climate (Peters & Herrick, 2002).

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